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Post-fire regeneration in alpine heathland: Does fire severity matter?

JAMES S. CAMAC,[1,3]* RICHARD J. WILLIAMS,[4] CARL-HENRIK WAHREN,[3] WILLIAM K. MORRIS[1] AND JOHN W. MORGAN[2,3]

[1]*School of Botany, The University of Melbourne, Parkville, Vic. 3010, Australia (Email: james.camac@gmail.com)*, [2]*Department of Botany, La Trobe University,* [3]*Research Centre for Applied Alpine Ecology, La Trobe University, Melbourne, Victoria, and* [4]*CSIRO Ecosystem Sciences, Winnellie, Northern Territory, Australia*

Abstract Fire severity is thought to be an important determinant of landscape patterns of post-fire regeneration, yet there have been few studies of the effects of variation in fire severity at landscape scales on floristic diversity and composition, and none within alpine vegetation. Understanding how fire severity affects alpine vegetation is important because fire is relatively infrequent in alpine environments. Globally, alpine ecosystems are at risk from climate change, which, in addition to warming, is likely to increase the severity and frequency of fire in south-eastern Australia. Here we examine the effects of variation in fire severity on plant diversity and vegetation composition, 5 years after the widespread fires of 2003. We used floristic data from two wide-spread vegetation types on the Bogong High Plains: open heathland and closed heathland. Three alternative models were tested relating variation in plant community attributes (e.g. diversity, ground cover of dominant species, amount of bare ground) to variation in fire severity. The models were (i) 'linear', attributes vary linearly with fire severity; (ii) 'intermediate disturbance', attributes are highest at intermediate fire severity and lowest at both low- and high-severity; and (iii) 'null', attributes are unaffected by fire severity. In both heathlands, there were few differences in floristic diversity, cover of dominant species and community composition, across the strong fire severity gradient. The null model was most supported in the vast majority of cases, with only limited support for either the linear and intermediate disturbance models. Our data indicate that in both heathlands, vegetation attributes in burnt vegetation were converging towards that of the unburnt state. We conclude that fire severity had little impact on post-fire regeneration, and that both closed and open alpine heathlands are resilient to variation in fire severity during landscape scale fires.

Key words: competing model, disturbance, fire management, intensity, intermediate disturbance hypothesis, resilience, shrub.

INTRODUCTION

Fire influences the distribution, composition and structure of vegetation at global, regional and local scales (Bond *et al*. 2005) and therefore is integral to conservation management (Bradstock *et al*. 2002; Andersen *et al*. 2003; Bond & Archibald 2003). The spatial and temporal extent of fire is a function of fire regime (frequency, intensity, season, type), the variation in which can affect ecosystem state, ecological processes, landscape heterogeneity and biodiversity (Bradstock *et al*. 2002). The effects of fire-line intensity (the rate of energy release per metre of fire front) on ecosystems are difficult to investigate because quantifying intensity is rarely practicable. Fire severity is often used as a surrogate for fire intensity as it is more readily quantifiable and directly assesses the effects of fire on ecosystem attributes (Keeley 2009). Patterns of fire severity may vary considerably during individual fires, including large severe fires, even under extreme fire weather conditions (Knox and Clarke 2012). Some points in the landscape burn severely, others only slightly or not at all (Turner *et al*. 1999; Schoennagel *et al*. 2008; Williams & Bradstock 2008).

The effects of variation in fire intensity or severity on ecosystems is the subject of considerable public and scientific debate. Some have argued that in temperate forests and shrublands large, severe fires are an unnatural expression of the historical fire regime, and are ecologically destructive (e.g. Adams & Attiwill 2011; Minnich 2001). According to this view, such fires are ecologically damaging because of the resultant homogenization of landscapes, loss of fine-grained patchiness, the death of many individuals of both plants and animals and the destruction of refugia.

*Corresponding author.
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Conversely, others have shown that high severity fires in forests have few long-lasting effects on the vegetation (e.g. Keeley *et al.* 2008; Knox and Clarke 2012). These concerns regarding the role of large, potentially severe fires are emerging as major issues influencing the management of fire on public land in Australia, including National Parks, with calls for more prescribed burning to mitigate fire size and severity in the effort of maintaining biodiversity (e.g. Adams & Attiwill 2011).

While large, severe fires undoubtedly threaten life and property, they are not necessarily threats to biodiversity (Williams & Bradstock 2008). Indeed, in many temperate regions of the world, occasional high severity fires are increasingly seen as integral to the historical fire regime (Bradstock 2008; Schoennagel *et al.* 2008; Veblen *et al.* 2008). Ecological processes such as mortality (Keeley 2006; Ryan & Williams 2011), recruitment (Hodgkinson 1991; Moreno & Oechel 1991; Turner *et al.* 1999; Myers & Harms 2011), ecosystem productivity (Hodgkinson 1991; Turner *et al.* 2003) and the maintenance of alternative stable states (Odion *et al.* 2010) have all been shown to vary with fire severity. Fire severity may also affect competitive interactions among plants in the post-fire environment (Ducey *et al.* 1996), which may affect succession (Moreno & Oechel 1991; Turner *et al.* 1999, 2003) and diversity (Schoennagel *et al.* 2008). Variation in severity may thus provide a fundamental template for the maintenance of ecological processes that underpin biodiversity.

Here we examine how variation in fire severity affects structure, composition and diversity of alpine heathlands. Australian alpine landscapes are ideal ecosystems to test the above contrasting hypotheses concerning fire regimes and biodiversity because they are subject to recurrent albeit infrequent fire; and typically burn only under severe fire weather and widespread drought (Williams *et al.* 2006b, 2008). Fires arrive via the forested slopes and foothills and the alpine vegetation burns with varying severity (Williams *et al.* 2006b, 2008). The vegetation is also diverse, consisting of a mixture of plant communities and life-forms, and is relatively slow-growing. Moreover, fires in south-eastern Australia are projected to become more intense as a consequence of climate change because the number of days with severe fire weather is likely to increase (Cary *et al.* 2012). Given that alpine ecosystems globally are threatened by climate change (IPCC 2007; Engler *et al.* 2011), understanding how these systems respond to changes in fire regime will be critical in devising strategies that conserve global biodiversity.

We examine how the structure and composition of alpine heathland varies as a function of fire severity 5 years after landscape scale fires burnt more than 1 million hectares across south-eastern Australia in 2003. We focus on heathlands because they occupy about 60% of Australian alpine landscapes (Williams *et al.* 2006a) and shrubby vegetation is common in most alpine and high latitude areas of the world (Sturm *et al.* 2001). As fire is relatively infrequent in alpine areas (Körner 2003) and post-fire regeneration may be constrained by low temperatures, the effects of fire on alpine vegetation may be long-lasting, particularly if severity is high (Kirkpatrick *et al.* 2010). Alternatively, as shown for a range of lowland shrubby vegetation types (Keeley *et al.* 2008; Keeley 2009), the effects of fire on alpine heathlands may be short-lived and independent of fire severity.

In this post-fire study we examined whether (i) there was a difference in community composition and structure between heathlands burnt in 2003 and those that were unburnt in 2003, and (ii) whether composition and structure of burnt heathland varied with fire severity. We assessed how variation in fire severity at the landscape scale affected six vegetation attributes in open heathland and closed heathland: species density and diversity, shrub cover, cover of the dominant snow grasses (*Poa* spp.), cover of bare ground and species composition. All are important measures of landscape state in relation to disturbance in Australian alpine vegetation (Wahren *et al.* 1994; Williams *et al.* 2006a). For the second question, we proposed three alternative models relating variation in cover and diversity to variation in fire severity: (i) 'linear', where attributes vary linearly (either positively or negatively) with fire severity; (ii) 'intermediate disturbance' (following Huston 1979), where responses to variation in severity are non-linear, and are highest at intermediate levels of fire severity and lowest at both low- and high-severity; and (iii) 'null', where there is no detectable effect of fire severity on the measured attributes.

METHODS

Study area

Our study was conducted across the Bogong High Plains in the Alpine National Park (37°S, 147°E), about 250 km north-east of Melbourne, Australia. Altitude ranged from 1600 to 1894 m a.s.l. Mean annual precipitation is 1228 mm. Winter snow cover lasts from June to September with low average annual temperatures (mean minimum = 2.5°C; mean maximum = 9.3°C) and frequent frost (Australian Bureau of Meteorology, unpubl. data, 2011). Soils are organic loams, and the vegetation is a mosaic of Eucalypt-woodland and treeless vegetation, with the latter ranging from closed and open heathlands, herbfields, tussock grasslands and wetlands. We studied closed heathland and open heathland, which together occupy about half the study area. Closed heathland occurs on steeper slopes; shrub cover is 70–100%, and grass and herb cover is generally <10%. The dominant shrub species range from 1–2 m tall and include *Bossiaea foliosa* (Fabaceae),

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Orites lancifolia (Proteaceae), *Phebalium squamulosum* (Rutaceae) and *Prostanthera cuneata* (Lamiaceae). Open heathlands occur on gentle slopes, shrub cover is 20–50%, and 0.2–0.5 m tall. The dominant shrub is *Grevillea australis* (Proteaceae), with the inter-shrub spaces dominated by snow grasses (*Poa* spp.) and herbs (Williams *et al*. 2006a).

Fires in Australia's south-eastern highlands in 2003 coincided with Australia's longest drought since 1900. Approximately 50% (400 000 ha) of the Alpine National Park was burnt, including about 10 000 ha of alpine treeless vegetation. Occurrence and severity varied substantially, with the scale of burning-induced patchiness ranging from square metres to square kilometres. Approximately 87% of closed heathland, 59% of open heathland and 13% of grassland vegetation was burnt (Williams *et al*. 2006b).

Study design

Immediately after the 2003 fires, we surveyed the patterns of burning (occurrence and severity where burnt) in four common vegetation types: grassland, snow patch herbfields, open heathland and closed heathland (Williams *et al*. 2006b). We established 419 randomly located survey points over an area of about 100 km^2. Latitude and longitude ($\pm$ 10 m), whether the point was burnt or unburnt by the 2003 fires, slope, aspect, altitude and vegetation type were recorded. Fire severity in heathlands was based on minimum twig diameter, a proxy measure of fire severity (Whight & Bradstock 1999). Twig diameter was recorded between 1 and 10 months post fire, from 10 replicate samples of the two dominant shrub species, if present: *Grevillea australis* in open heathland, and *Orites lancifolia* in closed heathland. Twig diameters ranged from about 1–21 mm, with smaller diameters assumed to have experienced lower intensity of fire. This encompassed the full range of fire severity, from unburnt and lightly scorched shrubs, with only the tips of the outer branches burnt, to severely burnt shrubs where the majority of even large branches were consumed. Some sample sites, including those unburnt in 2003, are likely to have been burnt in 1939. However, the fire history of the sites prior to 2003 is unknown, and henceforth 'unburnt' will refer simply to sites that were not burnt in the 2003 fires.

All heathland points surveyed in 2003 were relocated in 2007/08 and a subset selected for the floristic survey, according to the following criteria: sites had to (i) be greater than 0.25 ha in area, so that the site could be sampled adequately by a 50 m transect (see below); (ii) be at least 50 m from wetlands, to minimize the chance of wetland species affecting the species composition of the heathlands, and (iii) have a measure of minimum twig diameter from the dominant shrubs (not all heathland sites surveyed for fire occurrence immediately after the 2003 fires had concomitant measures of twig diameter; Williams *et al*. 2006b). This yielded 150 sites that encapsulated the full range of fire severity, including unburnt vegetation, in both heathlands. From these we chose 40 sites at random from each community, of which 30 were burnt and 10 were unburnt. Half the sites were in the northern part of the Bogong High Plains, which had been ungrazed by cattle since their removal in 1991; the other half of the sites were in the southern part of the Bogong High Plains, which were grazed by cattle at the time of the 2003 fires, and had been for over 150 years prior to that. Grazing ceased on the Bogong High Plains immediately after the 2003 fires, and has not recommenced. Grazing by domestic stock has long been a contentious issue in the ecology and management of Australia's alpine environments (Williams *et al*. 2006a) and this aspect of the design allowed us to test explicitly whether the effects of fire severity depended on grazing history. Average minimum twig diameter across burnt sites ranged from 4 to 21 mm in closed heathland, and from 2 to 13 mm in open heathland. The floristic composition at each sample site was determined using a single 50 m transect, along which five 6 m^2 quadrats were placed at 10 m intervals. Within each quadrat, the cover of all vascular plant species, total shrub cover and bare ground were estimated using the Braun-Blanquet cover abundance scale (Braun-Blanquet 1965). Difficult taxa, such as *Poa*, were identified to genus level, with the dominant grass in open and closed heathlands often being *Poa hiemata* and *Poa hothemensis*, respectively.

Data analysis

For all analyses, the transect was the experimental unit, with minimum twig diameter treated as a continuous variable. Braun-Blanquet cover data were converted to mid-point percent values. Average cover of each taxon at each site (transect) was then calculated from the five quadrats. Because minimum twig diameters are not directly comparable between open and closed heathlands (due to differences in stem architecture), diameters were standardized as a proportion of the maximum twig diameter found in each community. Sites in unburnt heathland were given a standard unburnt minimum twig diameter of 0.5 mm, allowing variation in the dependent variables to be presented as a function of twig diameter on the same scale for each community.

Data were analysed in two stages. First, to determine the difference between burnt and unburnt heathland, we compared the mean value of each variable between unburnt and burnt sites (pooling fire severity measures), using 95% confidence intervals. If the 95% confidence intervals of the unburnt samples did not overlap with the intervals for the burnt samples, we inferred a significant difference for that attribute (Cumming & Finch 2005). Second, to assess how fire severity affected response variables, we modelled the relationship between standardized minimum twig diameter and the mean site attribute in burnt heathland. Unburnt values were excluded from this second set of analyses because we were explicitly interested in discriminating between the three models only for burnt heathland. Including the unburnt data would also result in the independent variable being non-continuous between minimum and maximum values. We used general linear models with the 'glm' function from the statistical software package R (R Development Core Team 2011). To model cover, which is bound within the unit interval, data were logit transformed. We assumed normal error distributions for the density and diversity data. Though technically both of these community measures are bound at zero, neither approached this limit. For the null models, we fitted only an intercept term, that is,

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without a parameter that accounted for a relationship with standardized minimum twig diameter. For the 'linear' models, we fitted both an intercept term and a slope term describing the relationship with standardized minimum twig diameter. For the 'intermediate disturbance' models, we fitted second order polynomial terms, which included a parameter for standardized minimum twig diameter and an additional parameter for its square. This was used because it can reflect the theoretical shape of the intermediate disturbance hypothesis. To display the uncertainty of model fits, we used an approximate Bayesian approach to construct credible intervals. We simulated posterior densities, based on parameter point estimates and standard errors returned by the 'glm' function, using the 'sim' function of the R package 'arm' (Gelman & Hill 2007). From these simulated densities, we used the inner 95% quantiles as approximate credible intervals. To compare alternative models and their corresponding hypotheses, we used an information theoretic approach based on Akaike's Information Criteria (AIC). Models having AIC values within 1–2 of the minimum were considered to have equal support; values within 4–7 of the minimum had considerably less support than the minimal model (Burnham & Anderson 2002). Where linear and intermediate disturbance models received support, the fitted models were used to calculate predicted values of the response variables as a function of standardized twig diameter.

To assess floristic compositional changes in relation to fire severity and grazing history, we used non-metric multidimensional scaling (NMDS) with the Bray–Curtis dissimilarity index based on both the presence/absence of species as well as their cover data for each transect. The contribution of environmental variables (fire severity, grazing history, altitude, slope) to the ordination pattern was assessed using the ADONIS function of the 'vegan' package (Oksanen *et al.* 2012). ADONIS is an alternative, more flexible permutation procedure to ANOSIM (e.g. Clarke 1993) that partitions the sums-of-squares of the Bray–Curtis dissimilarity matrix (Anderson 2001; Oksanen 2011).

RESULTS

Five years after fire, there was substantial regeneration in both closed and open heathlands. Total vegetation cover was generally >80%, with prolific growth and flowering of snow grasses, and substantial regeneration of shrubs. In closed heathland, 75 species were recorded. Species density ranged from 3–24 per 6 m^2 quadrat, and 11–33 per 50 m transect. In open heathland, there were 78 species and species density ranged from 7–26 per quadrat and 18–36 per transect. There were some differences in the response variables between burnt and unburnt heathland, but the differences were generally slight to modest and not consistent for most variables (Table 1). Mean species density per transect was similar in burnt and unburnt vegetation in both closed heathland and open heathland. Shannon diversity was higher in burnt closed heathland compared with unburnt closed heathland, whereas in open heathland it was identical in both burnt and unburnt vegetation. Shrub cover was substantially lower in burnt compared with unburnt closed heathland, whereas the difference was more muted in open heathland. The cover of snow grass (*Poa* spp.) was substantially higher in burnt compared with unburnt closed heathland, but in open heathland, *Poa* cover was similar in burnt and unburnt samples. In both communities, the amount of bare ground in unburnt samples was substantially lower than that in burnt areas.

Species density, Shannon diversity and *Poa* cover were equally supported by all models (null, linear and intermediate disturbance) in both closed and open heathlands (Figs 1 and 2 respectively). In all cases the explained deviance was low (<13%). For shrub cover in closed heathland, the intermediate disturbance model had the most support (Δ AIC = 6; 22% of the deviance explained; Fig 1). The peak average shrub cover (about 30%) occurred for a standardized twig value of 0.585; this was about 11% greater than the expected shrub cover at the lowest twig diameters, and 13% greater than shrub cover at the highest twig diameters. In open heath, the effect of fire severity on open heath shrub cover was equally supported by all models (Fig. 2). For bare ground, the linear and intermediate disturbance models had equal support relative to the null model in open heathland (ΔAIC = 8 and 4, respectively; <30% of the deviance explained; Fig. 2). In this case, the linear model (model with greatest support) estimated average bare ground cover was 30% greater at high twig diameters than at low twig diameters. For bare ground in closed heathland, the

Table 1. Mean (±95% CI) vegetation attributes responses to burning in closed and open heathlands, 5 years post fire

Attribute (mean ± 95% CI)	Closed heath (burnt)	Closed heath (unburnt)	Open heath (burnt)	Open heath (unburnt)
Species density	22.1 ± 2.2	19.1 ± 2.9	26.9 ± 2.0	29.6 ± 2.5
Shannon diversity	2.1 ± 0.2	1.5 ± 0.2	1.8 ± 0.2	1.8 ± 0.2
Shrub cover, %	25.3 ± 2.7	81.5 ± 3.1	8.8 ± 1.2	14.2 ± 3.7
Poa cover, %	14.2 ± 2.4	1.7 ± 1.0	32.2 ± 6.3	33.2 ± 4.3
Bare ground, %	16.2 ± 1.6	2.0 ± 0.4	22.6 ± 2.1	2.0 ± 0.4

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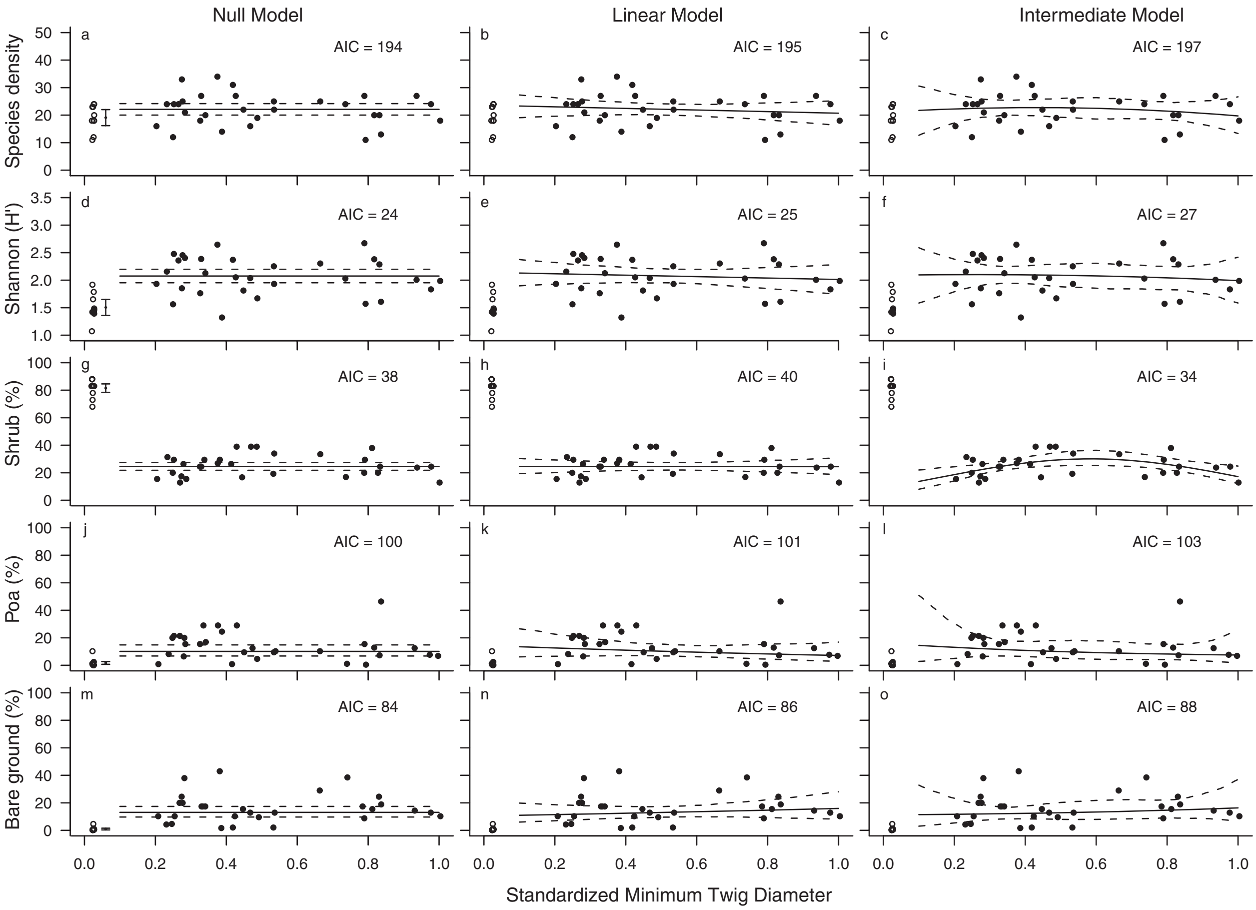

Fig. 1. Three competing models of fire severity effects on species density (a–c), Shannon diversity (d–f), and the cover of shrubs (g–i), *Poa* (j–l) and bare ground (m–o) in closed heathland. Left panels are null models (no severity effect); centre panels are linear models (positive or negative effects); right panels are intermediate disturbance models (greatest effects occurs at intermediate severity). Open circles on left side of figures signify unburnt sites, filled circles are burnt sites. Error bar in null model panels = mean from unburnt sites (±95% CI). Model fit is based on burnt sites and is illustrated by a solid line, dashed lines are 95% credible intervals. AIC values are shown. AIC, Akaike's Information Criteria.

null model had the most support. Grazing history had no significant effect on diversity and life-form cover, and there were no significant severity x grazing interactions.

Ordination based on cover data found no significant differences between burnt and unburnt heathland and no significant correlation with fire severity (closed heathland: three-dimensional stress = 0.16, $R^2 = 0.05$, $P = 0.99$; open heathland: two-dimensional stress = 0.15, $R^2 = 0.07$, $P = 0.99$; Fig. 3) or any of other environmental variables (grazing history, altitude, slope, aspect or any interaction; $P = 0.99$, $R^2 < 0.09$, for all variables and interactions). Ordinations based on presence/absence data gave very similar results.

DISCUSSION

We proposed and tested three competing models for the impact of fire severity on post-fire regeneration in alpine heathlands, finding strong support for the null model. Five years after landscape-scale fire, there were few differences in plant diversity and floristic composition across a strong fire severity gradient. Of the six measures of vegetation state assessed, the null model was most supported for all but two: shrub cover in closed heathland and bare ground in open heathland. In each case the model explained less than 30% of the deviance. Furthermore, ordination showed no difference between burnt and unburnt vegetation, nor was there a significant effect of fire severity on species composition in either burnt heathland community. Thus, 5 years post fire, the floristic composition of both alpine heathlands had converged towards that of the unburnt state. In Australian alpine and subalpine vegetation, most species re-establish within one or two years after fire (Wahren *et al.* 2001; Walsh & McDougall 2004), suggesting that these heathlands, like many temperate sclerophyllous shrubby vegetation types, follow the 'initial floristic composition' model of Egler

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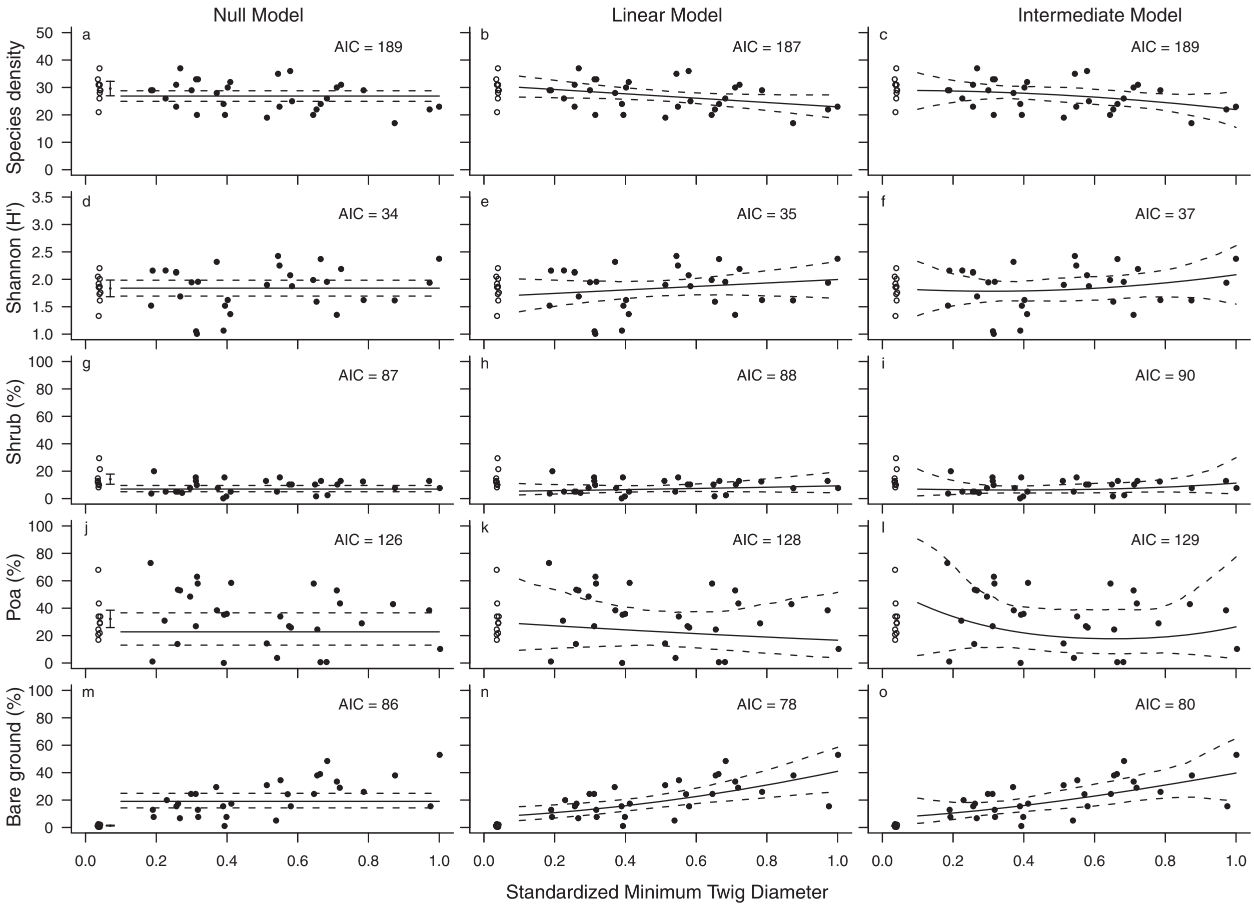

Fig. 2. Three competing models of fire severity effects on species density (a–c), Shannon diversity (d–f), and the cover of shrubs (g–i), *Poa* (j–l) and bare ground (m–o) in open heathland. Left panels are null models (no severity effect); centre panels are linear models (positive or negative severity effects); right panels are intermediate disturbance models (greatest effects occur at intermediate severity). Open circles on left side of figures signify unburnt sites, filled circles are burnt sites. Error bar in null model panels = mean from unburnt sites (±95% CI). Model fit is based on burnt sites and is illustrated by a solid line, dashed lines are 95% credible intervals. AIC values are shown. AIC, Akaike's Information Criteria.

(1954) or the 'auto-succession' model of Keeley *et al.* (2005), whereby species diversity and composition is established rapidly post fire.

Post-fire regeneration in shrubby vegetation, whether vegetative or via seedlings, may be influenced by fire severity (Moreno & Oechel 1991; Bond & Midgley 2001; Myers & Harms 2011) which, in turn, could lead to different post-fire composition and diversity. However, we found no evidence that variation in fire severity affected community diversity and composition 5 years after fire. The muted effect of fire severity on compositional measures, and relatively rapid return to a pre-fire state, has also been documented in lowland shrubby vegetation types. Morrison (2002) found that fire intensity explained only 10% of the floristic variation in lowland Australian heath. Similarly, Knox and Clarke (2012) showed no effect of fire severity on fire-cued species in a temperate forest, and Knox and Clarke (2011) showed little long-term effect of fire severity on woody plant resprouting ability. In Californian chaparral fire severity has been shown to have little long-lasting effects on vegetative regeneration (Keeley *et al.* 2005, 2008). This independence of post-fire regeneration patterns and fire severity appears to be very common in temperate heathlands. However, fire severity may interact strongly with other factors, such as productivity, rainfall and soil moisture to determine post-fire patterns of plant diversity (Safford & Harrison 2004; Pausas *et al.* 2008; Myers & Harms 2011).

The muted effect of fire severity on diversity and composition in alpine heathlands is likely due to the strong resprouting or seeding capacity of the Australian heathland floras in general (Williams *et al.* 2006a; Enright *et al.* 2012), and the long evolutionary history of recurrent fire in temperate shrublands worldwide (Keeley *et al.* 2011). Australian alpine heathlands are dominated by genera that are common in lowland

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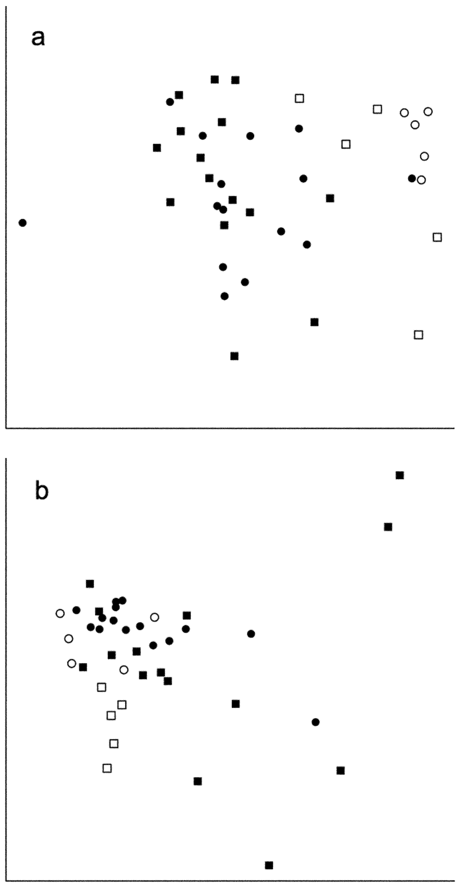

Fig. 3. Non-metric multidimensional scaling (NMDS) of floristic samples, showing. (a) closed heathland (showing the first two axis from three-dimensional solutions – minimum stress 0.16); (b) open heathland (two-dimensional, minimum stress 0.15). Configurations were based on relative species abundance. Burnt/grazed = black circle; burnt/ungrazed = black square; unburnt/grazed = open circle; unburnt/ungrazed = open square.

heathlands that have evolved in conjunction with recurrent fire over millions of years, with variation in fire intensity (and by implication, severity) being an integral part of fire regimes in Australian heathlands (Keith *et al.* 2002). Fires, of course, may result in substantial amounts of bare ground and although levels of bare ground are naturally low (1–2%) in long-unburnt alpine vegetation, the vast majority of the Australian alpine vascular flora is capable of recolonizing bare ground gaps that result from frequent small-scale disturbances associated with frost heave and insect attack (Williams *et al.* 2006a). Moreover, Australian alpine shrubs require bare ground for successful seedling recruitment. Thus, the patterns of regeneration in response to infrequent, large-scale and potentially severe fire rely upon the same traits of the Australian alpine flora that has allowed it to persist in the face of recurrent, frequent small-scale disturbances.

Climate change has already affected mountain ecosystems worldwide (e.g. Sturm *et al.* 2001; Chapin *et al.* 2005), while both severity and frequency of fire have been predicted to increase in many temperate ecosystems of the world (Flannigan *et al.* 2009), including Australian heathlands (Enright *et al.* 2012). Our data suggest that Australian alpine heathlands, even though they may be extensively burnt during occasional large fires, are resilient to variation in severity. Therefore, individual high severity fires, a feature of the current and future fire regimes of south-eastern Australia, are unlikely to result in the local extinction of alpine heathland species or communities (Williams *et al.* 2008). However, Australian heathlands more generally are known to be sensitive to variation in fire return-interval (Enright *et al.* 2012) and an increase in the frequency of fire may threaten alpine heathlands. Given that dominant obligate-seeding shrubs of the Australian alps take at least 5 years to flower and produce seed (C.-H. Wahren, unpubl. data, 2012), then detrimental intervals between fires are likely to be of the order of 10 years or less.

Our results have important implications for the conservation management of alpine ecosystems, under both current and future climate. Fire in some alpine environments may have long-lasting effects on some vegetation types, such as coniferous heaths in Tasmania (Kirkpatrick *et al.* 2010), and hygrophilous shrubs that occur in *Sphagnum* wetlands (McDougall 2007). However, our data show clearly that a single fire, albeit large and severe, did not reduce diversity or substantially change the composition of two of the most common alpine vegetation types in Australia – closed heathlands and open heathlands. Together these vegetation types account for over 60% of the alpine landscape. Recurrent, infrequent and large fires that result in parts of the alpine landscape being burnt severely are a part of the historical fire regime of the Australian Alps (Williams *et al.* 2006a, 2008). Propagation of these fires across the landscape is highly dependent on these two vegetation types (Williams *et al.* 2006b) but as our data show there are few long-lasting effects on diversity and composition in these heathlands when burnt, even if burnt severely.

Our data indicate there is no conservation imperative to limit fire severity in alpine landscapes through active fuel management in alpine vegetation (e.g. by prescribed burning or livestock grazing) based on the hypothesis that large, severe fire is unnatural and has deleterious impacts on biodiversity in southern Australia (e.g. Adams & Attiwill 2011). Although diversity and composition of the two-heathland communities were largely unaffected by fire severity, vegetation structure was. Bare ground was substantially higher in burnt vegetation than in unburnt, in both open and closed heathlands. Shrub cover was substantially lower in burnt closed heath than unburnt closed heath. At

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least a decade post fire is likely to be required before the cover of dominant shrubs and the amount of bare ground return to pre-fire levels. Dense, mature shrub cover is required as habitat for a number of specialist alpine faunal species (Sanecki *et al.* 2006), and elevated levels of bare ground (>5%) increase the risk of soil loss in Australian alpine vegetation (Williams *et al.* 2006a, 2008). Thus, minimizing further disturbance to alpine heathlands that are regenerating post fire should be a fundamental management objective.

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